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Losses of salt marsh in China: Trends, threats and management

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ABSTRACT

Coastal salt marsh, one of the blue carbon ecosystems that can adapt and mitigate climate change influence, is drawing global attention due to its high carbon sequestration capability. In China, however, coastal salt marsh has suffered great losses. Nation-wide analysis of salt marsh trends and management is critical to ecosystem protection and restoration. Thus, by analyzing previous coastal salt marsh studies, we found that the extent of coastal salt marsh varied greatly among the Liao River Delta, the Yellow River Delta, the middle coast of Jiangsu Province, Chongming Dongtan and Jiuduansha in Shanghai, with a 59% overall loss of salt marsh extent from the 1980s to the 2010s. The rate of salt marsh loss slowed down after the year 2000. Coastal land-claim (reclamation) is the most dominant driver of salt marsh loss. Climate change and coastal erosion, invasive species, and vegetation dynamics driven by competition and succession have also led to various effects on salt marsh extent and the ecological services they provide. Sea level rise, reclamation pressure and environmental pollution are the main factors, as negative drivers, together with conservation and restoration policies, as positive ones, affecting future trends in salt marshes. China has implemented several measures to protect and restore salt marshes, such as setting up protected areas, drawing marine ecological redline, and making strict regulations on reclamation. However, stronger legal protection for wetlands, more effective enforcement, and participation by local communities can further enhance salt marsh restoration, conservation and management.

1. Introduction

Coastal salt marshes are ecosystems dominated by salt-tolerant herbaceous plants or shrubs that extend along the upper intertidal area of sedimentary coasts in all continents, except Antarctica, particularly in large river deltas and estuaries with sedimentary shorelines. Salt marshes provide considerable coastal and flood risk management benefits, as they can reduce wave energy as natural buffer zones in front of tidal defenses (Möller et al., 2014). Moreover, they are of great value to wildlife, since they support habitats and species of national and international significance (Phelan et al., 2011). Specifically, salt marsh ecosystems are highly productive providing breeding, feeding and overwintering sites for a large number of species, particularly birds (Hobbs and Shennan, 1986; McCurdy, 1988; Goss-Custard and Yates, 1992), including endangered species, such as the Red-crowned Crane (*Grus japonensis*). Additional demonstrable benefits of salt marsh ecosystems include water purification and shoreline protection through wave attenuation and sediment accretion, and a high capacity for carbon sequestration and storage (Chmura et al., 2003). Hence, salt marsh ecosystems have high intrinsic conservation value.

Unfortunately, salt marsh ecosystems have often been claimed (reclaimed) to support economic development, such as agricultural use, aquaculture ponds, industrial land and urban expansion (McCurdy, 1988; Gedan et al., 2009), leading to a large global decline in salt marsh area (Duarte et al., 2008). These losses also contribute to the component of greenhouse gas emissions derived from land use change, due to the loss of CO_2 sink capacity, the emissions as CO_2 of organic carbon stored in salt marshes following disturbance, and the emissions of methane when hydrological flows are discontinued (Altor and Mitsch, 2008). Hence, salt marshes are recognized as key blue carbon ecosystems, whose conservation and restoration can be effective components of climate change mitigation and adaptation strategies (McLeod et al., 2011; Duarte et al., 2013). Indeed, salt marsh ecosystems have the broadest spatial distribution among three main blue carbon ecosystems (salt marsh, mangrove and seagrass), ranging from temperate to subtropical zone, and supporting the highest rates of carbon burial rate per unit area (McLeod et al., 2011; Duarte et al., 2013). Additionally, it is expected that sea level rise with climate change may lead, where

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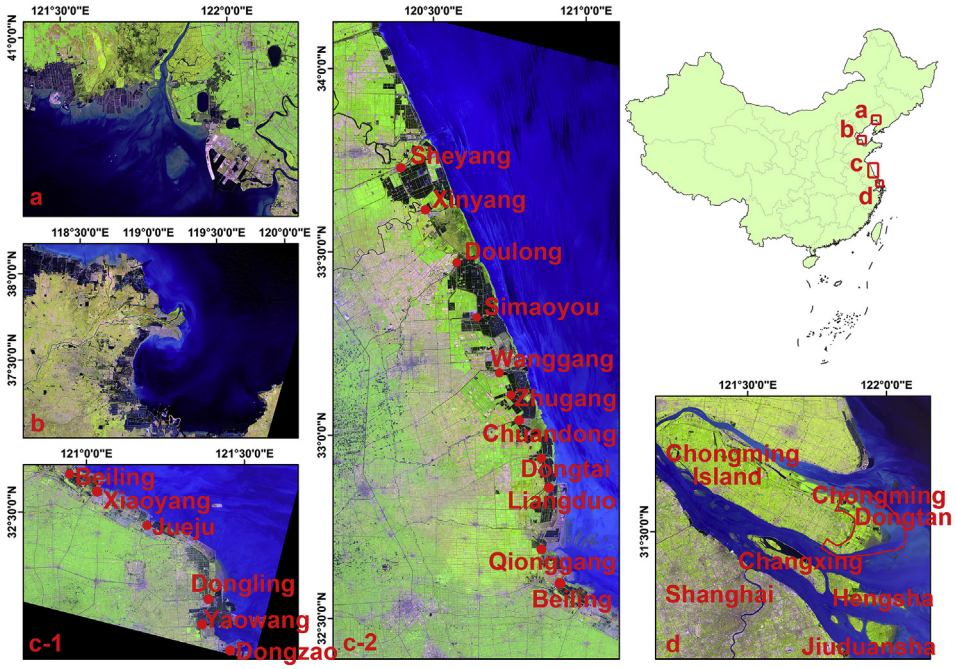

Fig. 1. Locations of dominant salt marsh areas along coastal China (Landsat 8 Operational Land Imager data from the United States Geological Survey; *a*, the Liao River Delta in Liaoning Province; *b*, the Yellow River Delta in Shandong Province; *c*, different coastal sites in the middle coast of Jiangsu Province; *d*, Chongming Dongtan, Changxing Island, Hengsha Island and Jiuduansha in Shanghai City). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

sediment supply is adequate, to increased salt marsh carbon burial and accretion rates during the first half of the 21st century (Donato et al., 2011; Duarte et al., 2013). However, Crosby et al. (2016) suggested by the end of the 21st Century that 60%–91% of the salt marshes would not keep pace against IPCC-predicted rates of sea level rise.

The rigorous assessment of the distribution and magnitude of salt marsh loss provides fundamental information for restoration and protection purposes within blue carbon strategies. Changes in salt marsh area are related to natural processes as well as direct and indirect anthropogenic activities. Total sediment inputs contribute to changes in salt marsh extent (Wang et al., 2014a), with high sediment supply providing opportunities for seaward growth. Variations in physical properties, such as salinity, nutrients, and water supply can also lead to relative changes in salt marsh extent (Odum, 1984). Conversely, salt marsh vegetation contains ecosystem engineering species, themselves influencing soil and sediment characteristics (Zhou et al., 2007) as well as their spatiotemporal variation (Zhao et al., 2015). Sea level rise poses already a threat of drowning to salt marsh ecosystems in many places (Reed, 1995; Kearney et al., 2002; Carniello et al., 2009). Only salt marsh ecosystems with high sediment supply are expected to benefit from sea level rise over the coming decades, but even those may be drowned if sea level rise rate accelerates (Stralberg et al., 2011; Wang et al., 2014a; Ge et al., 2016). In addition to environmental factors, biotic factors also lead to changes in salt marsh extent, such as competitive exclusion and facilitation by either other plants or animals, whether native or exotic (Bertness, 1991; Wang et al., 2006). Anthropogenic activities can affect salt marsh extent significantly, with reclamation being a main driver of the global loss of salt marsh ecosystems (An et al., 2007; Gedan et al., 2009; Kennish, 2013; Du et al., 2016), while protection and restoration increasing salt marsh extent (Broome et al., 1988; Zedler, 1988; Warren et al., 2002; Wolters et al., 2005), although at a slower pace.

The realization of the important role of salt marshes, particularly in climate change mitigation and adaptation, has led to salt marsh conservation and restoration programs in many nations, such as the Netherlands (Eertman et al., 2002) and the United Kingdom (Garbutt and Wolters, 2008). These programs require, however, detailed knowledge of the historical distribution of salt marshes, which is lacking in many regions of the world (Chmura et al., 2003), including China and South America, to help guide the development of feasible and sustainable restoration goals (Van Dyke and Wasson, 2005). A first global assessment of salt marsh distribution, based on data derived from different research articles, has now been made available (McOwen et al., 2017). However, as salt marsh data were collected from various periods, the corresponding map of salt marsh extent could only display areas where salt marsh had been documented in previous studies. While many salt marsh studies have been conducted in China, they are typically focused on local study areas, such as river deltas and natural reserves. Hence, an assessment of the status and trends of salt marshes at the national or regional-scales is still needed in China.

In view of the above considerations, here we present a quantitative review of salt marsh losses and changes in areas along coastal China where salt marshes have been assessed and we identify the key drivers involved. In particular, (*a*) based on published books/reports and journal papers, we review the salt marsh areal changes from north to south along the coast of China, describe the processes underlying these changes and attribute them to major drivers; (*b*) we discuss the factors associated with future changes in salt marsh extent, and (*c*) we review existing management practices to provide our recommendations toward a national plan to promote salt marsh restoration and conservation.

2. Methods

2.1. Study areas

China has a coastline extending approximately along 18,000 km, encompassing temperate, subtropical and tropical zones, all suitable to support coastal salt marshes, particularly in temperate and subtropical zones. In tropical shorelines, salt marshes may be replaced by mangroves. In addition, coastal shores in the north of Hangzhou Bay consist of intertidal mud/sand flats that may be potential places of coastal salt marshes, whereas those in the south of Hangzhou Bay mainly consist of rocky shores (Zhao and Wang, 2000). Thus, coastal salt marshes in China mostly concentrate in the north of Hangzhou Bay, except of some small areas of estuaries in the south of Hangzhou Bay for *Spartina alterniflora* (Wang et al., 2015). In the north of Hangzhou Bay, intertidal mudflats occur in the Liao River Delta, the Yellow River Delta, Laizhou Bay and the Yangtze River Delta (Jiangsu coast, Chongming Dongtan and Jiuduansha). Therefore, we selected these regions, except Laizhou Bay due to limited research data, as the study areas to investigate

Table 1
Salt marsh dynamics in Liao River Delta from 1988 to 2009.

Year	Salt marsh area (ha)	Change proportion (%)	Change rate (ha·y^{-1})
1988	11266	–	–
1995	5295	−53.0	−853
2000	3067	−42.1	−492
2004	663	−78.4	−600
2007	1215	83.2	184
2009	1241	2.1	12

(Here salt marsh refers to *Suaeda* dominated marsh. Data from Jia et al., 2015).

coastal salt marsh dynamics in China (Fig. 1). We believe these regions represent most of the salt marsh area in China, with the area not included by the available studies likely representing a minor fraction of the total salt marsh area (Supplementary Figure).

2.2. Material sources

We collected the existing data/information on salt marsh extent and dynamics from published reports, books and journal papers. Sources were identified by conducting a Web of Science search for English references and a CNKI search for Chinese references between October 2016 and November 2017 using the following search terms: (salt marsh or *Spartina alterniflora* or *Phragmites australis* or *Suaeda salsa* or wetland) and (loss or decline or variation or change or dynamic or distribution) and (China or the Liao River Delta or the Yellow River Delta or the Yangtze River Estuary or Jiangsu or Shanghai or Chongming Dongtan or Jiuduansha). Only sources containing specific data/information on salt marsh areal extent or dynamics met our standards for analysis (Supplementary Table). Data were sorted by region and time series and tabulated and graphically presented to illustrate salt marsh changes in different regions.

3. Coastal salt marsh changes/degradation

Salt marshes are present from the north (Liaoning) to the south (Guangxi) of the Chinese coastal region, mostly dominated by *Phragmites australis*, *Suaeda salsa*, *Scirpus mariqueter* and *Tamarix chinensis* among native salt marsh species and concentrated mainly in the Liao River Delta, the Yellow River Delta, the middle coast of Jiangsu Province, Chongming Dongtan and Jiuduansha. Among these four native salt marsh species, *Phragmites australis* has the broadest distribution in China. *Phragmites australis* grows in coastal salt marshes and inland freshwater marshes, with salt marsh habitat comprising half of the area covered by *Phragmites australis* in China. Most coastal *Phragmites australis* is concentrated in Liaoning Province, where nearly half of its total extent in coastal China is found. *Phragmites australis* is also abundant in Hebei and Jiangsu Provinces, contributing 30% of the area occupied by *Phragmites australis* in coastal China. Shandong and Shanghai also have significant areas covered by *Phragmites australis* (Wu and Cai, 1993). *Spartina anglica* and *Spartina alterniflora* are exotic salt marsh species introduced from England in 1963 and from North America in 1979, respectively, to support land reclamation due to their high capacity for sediment accretion (Wan et al., 2009). Now, only *Spartina alterniflora* is present across significant areas.

Coastal salt marshes in China started to be exploited in the 11th century (Wang et al., 2014b), and have suffered a great loss since the mid-20th century (Li et al., 2017). After the 1980s, most published salt marsh data were identified using Landsat series images, which had moderate spatial and spectral resolutions to delineate salt marsh. Thus, here we present salt marsh losses during the past three decades in dominant regions from north to south along coastal China (Fig. 1). Changes in the salt marsh extent are associated with a few main drivers, including coastal reclamation and infrastructural construction, climate change and other factors leading to coastal erosion, and vegetation dynamics driven by invasive species, competition and succession of salt marshes, which are presented in Section 3.1, 3.2, 3.3, and 3.4, respectively. Section 3.5 summarises salt marsh changes in the above regions.

3.1. Coastal reclamation and infrastructural construction

With the rapid population growth in China, the high demand for coastal land has led to reclamation of vast areas covered by salt marshes since the 1950s. Indeed, coastal provinces only occupy 13% of total area of China, but account for more than 50% of large cities, 40% of medium-sized and small cities, 42% of population as well as more than 60% of GDP (Xu, 2011; Luo, 2016). As a result, about 8,010,000 ha equivalent to 58% of coastal wetlands, including swamp, salt marsh, estuary, gulf and mangrove, were lost between 1950 and 2014 (Sun et al., 2015b), whereas at least 708,000 ha of salt marsh were lost to reclamation by the year 1990 (Yang and Chen, 1995; Qiu, 2012). For example, Jiangsu coastal region experienced a loss of 144,400 ha salt marsh area from the 1950s to late 1990s. Tidal wetlands along the Yellow Sea decreased by 70% due to reclamation over past 50 years (Murray et al., 2014). Below, we will present four major areas from north to south along coastal China to describe salt marsh losses caused by reclamation.

3.1.1. Salt marsh dynamics in the Liao River Delta, Liaoning Province

From 1976 to 2004, salt marshes in the Liao River Delta, Liaoning Province decreased by 45% (Huang, 2011). A large-scale reduction in *Suaeda* occurred from 11,266 ha in 1988 to 663 ha in 2004. Thereafter, *Suaeda* in the delta increased during 2007 and 2009 due to conversation and restoration (Table 1) (Jia et al., 2015). Losses of salt marshes resulted in fragmentation. For example, while 29 patches of *Suaeda heteroptera* Kitag covered 5077 ha in 1976, an area of only 682 ha remained by 2004 that was composed of 67 small patches (Huang, 2011). The high loss of salt marshes since the late 1980s was mainly due to reclamation (Table 1) (Jia et al., 2015). Increased food demands with rapid population growth led to reclamation of large areas of salt marshes to be converted to paddy rice fields (a conversion of 729 ha of salt marshes into paddy fields took place between 1988 and 2000), and aquaculture ponds (a conversion of 2655 ha of salt marshes into aquaculture ponds took place between 1988 and 2000) (Jia et al., 2015). Additionally, oil field developments contributed to substantial losses of natural wetlands including salt marshes. For example, during 1991–1995, a total of 31,850 ha wetlands were reclaimed in this province (Liu et al., 2000; Wang and Li, 2006).

3.1.2. Salt marsh dynamics in the Yellow River Delta, Shandong Province

Human activities also led to significant salt marsh loss in the Yellow River Delta (Shandong Province). Reclamation activities contributed to 40% of tidal flat loss and 35% of salt marsh loss between 1990 and 2008 (Ma et al., 2015). From 1984 to 2014, the area covered by *Suaeda salsa* was reduced by 78% (Liu et al., 2015), from 90,200 ha in 1984 to 64,430 ha in 1994, 27,740 ha in 2004 and 19,690 ha in 2014, a serious reduction that was concurrent with the rapid increase in coastal reclamation in this region. The area reclaimed for aquaculture, ports, seawalls, oil fields and salt pans grew from 12,350 ha in 1994, to 30,490 ha in 2004 and 47,960 ha in 2014, respectively (Liu et al., 2015). Losses of *Phragmites australis* salt marsh area were mainly due to the conversion into salt pans and aquaculture ponds between 2001 and 2005 and into channels between 2005 and 2010, resulting in a decline in the area covered by *Phragmites australis* from 19,800 ha in 2001 to 13,820 ha in 2005 and 9880 ha in 2010 (Sun et al., 2016). Mixed *Phragmites australis-Suaeda salsa-Tamarix chinensis* were mostly affected in the conversions between salt marshes and dry lands (Huang et al., 2012). Salt marsh conversion to dry land was mainly due to a combination of cultivation, oil exploration and salt pans. Conversely, dry land conversion to salt marsh was the result of coastal erosion and seawater

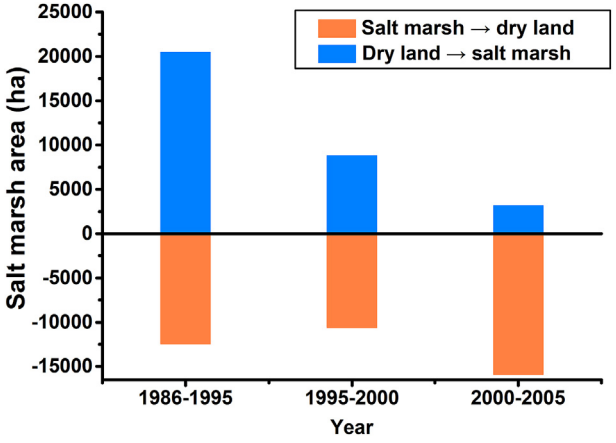

Fig. 2. Conversion between salt marsh and dry land in the Yellow River Delta (Data from Huang et al., 2012). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

intrusion (Huang et al., 2012). Reduction in freshwater and sediment inputs caused by dams upstream and groundwater extractions lead to coastal subsidence, leading to coastal erosion and seawater intrusion (Zhang and Li, 2008). Before 1995 the area of dry land converted into salt marsh exceeded that converted from salt marsh to dry land (Huang et al., 2012). However, this situation changed after 1995, leading to increasingly severe losses of salt marsh area over time. Thus, from 1986 to 2005, a total of 39,000 ha of salt marshes were converted into dry lands and 32,500 ha of dry lands were converted into salt marsh, leading to a net 6500 ha reduction of salt marshes (Fig. 2) (Huang et al., 2012).

3.1.3. Salt marsh dynamics in the middle coast of Jiangsu Province

Salt marsh loss in the middle coast of Jiangsu Province also resulted from land reclamation starting in the 1950s. The percentage of reclamation area affecting salt marsh ecosystems varied from 43.9% to 98.9% during each decade (Fig. 3) (Shen et al., 2006). Whereas areas for reclamation were mainly occupied by *Imperate cylindrical*, *Suaeda salsa* and mudflat between the 1950s and the 1990s, *Spartina alterniflora* marshes became the dominant resource for reclamation after the year 2000 (Fig. 3). Spatially, along the middle coast of Jiangsu, salt marsh loss at different locations varied from 980 ha to 8180 ha (Fig. 4), with the percentage of salt marsh loss by reclamation ranging from 36.4% to 92.8% with an average of 85.4% (Sun et al., 2015a). About 90% of salt marsh was lost in Doulong-Simaoyou, Sheyang-Wanggang, Chuandong-Dongtai and Liangduo-Qionggang regions (Fig. 4). Four consecutive land reclamations led to a severe reduction in salt marsh area, involving a total loss of 7410 ha area of salt marsh in Sheyang-Wanggang region, with the average width of salt marsh seaward from the progressively advanced dikes decreasing from 9 km to 5 km (Wang and Gao, 2005). This reclamation also led to a decrease of plant diversity in salt marshes, with *Spatina* and *Suaeda salsa* being the only species presented in large areas after the year 2000 (Zhang, 2013).

While great losses of salt marsh through the land reclamation in this region have occurred since the 1950s, significant new areas of salt marsh have evolved and expanded due to the high deposition of sediments from the Huai River basin, which substantially compensated the losses (Sun et al., 2015a). As a result, the net loss of salt marsh in this region is less than that of Liao River Delta or the Yellow River Delta.

3.1.4. Salt marsh dynamics in Chongming Dongtan, Shanghai City

In Chongming Dongtan, Shanghai City, the loss of salt marshes by land reclamation was traced to the establishment of state-run farms in the 1950s due to specific socio-economic conditions (Fig. 5). It involved a total loss of 20,814 ha salt marsh to 29 different reclamation events by 1990 (Shanghai Farm Reclamation Compilation committee, 2004). Large scale reclamation of salt marsh in this area was initiated during 1966 in Dongwangsha. From 1974 to 1981, a total of 750 ha salt marsh was reclaimed, with an annual average area of 107 ha. It increased to an annual average area of 187 ha reclaimed between 1981 and 1989, a total of 1497 ha during this period. The pace of reclamation increased sharply between 1989 and 2001, with an annual average of 937 ha conducive to a loss of 11,245 ha in this period. In parallel, two large reclamations took place in this period, and caused great salt marsh loss, including Dongwangsha reclamation with the constructions of Levee 1992 and Levee 1998, and Tuanjiesha reclamation with the construction of Levee 2001 (Liu et al., 2010) (Fig. 5). Specifically, a total of 6077 ha salt marsh were lost to reclamation after the construction of Levee 1992. From 1996 to 1997, a total of 914 ha area was reclaimed in the northern part of the area. In 1998, Levee 1998 was constructed, and a total of 2313 ha area was reclaimed between Levee 1992 and Levee 1998. A smaller reclamation (Levee 2001) was conducted between 2000 and 2002, with a total of 610 ha reclaimed in the southern part of the area (Wang and Zhang, 2005; Gao and Zhao, 2006).

In Chongming Dongtan, *Phragmites australis*, *Scirpus mariqueter* and *Spartina alterniflora* are the dominant salt marsh species. *Phragmites australis* is concentrated in the southern part, whereas *Spartina alterniflora* mostly occupies the northern part, and *Scirpus mariqueter* mainly grows in lower flats of the eastern part. The reclamation activities led to changes in the extent and distribution of these species. The reclamations in Tuanjiesha and Dongwangsha were conducted in high tidal flat with an elevation of about 3.6 m above the mean sea level (Gao and Zhao, 2006). High tidal flat reclamations resulted in a sharp decline of the area covered by *Phragmites australis* (Fig. 6), from 6762 ha in 1990 to 969 ha in 1998, as well as a subsequent loss of *Phragmites australis* from 1033 ha in 2000 to 542 ha in 2003 in this region.

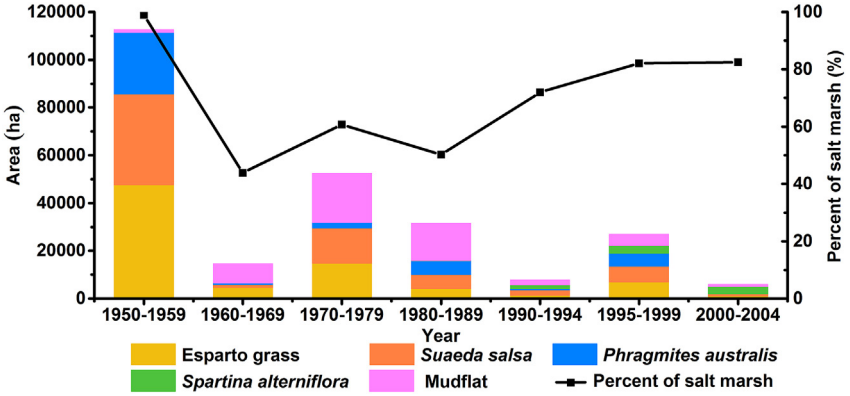

Fig. 3. Reclaimed salt marsh areas during different periods in the middle coast of Jiangsu Province (Left Y axis: reclaimed salt marsh areas; right Y axis: percent of reclaimed areas covered by salt marshes. Data from Shen et al., 2006).

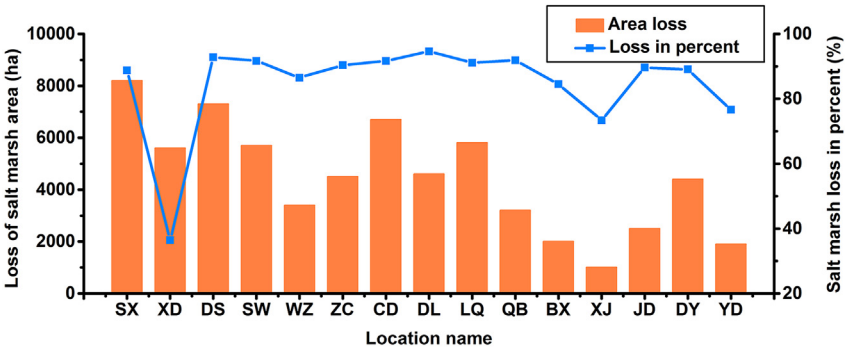
Area loss
Loss in percent
Loss of salt marsh area (ha)
Salt marsh loss in percent (%)
Location name
SX XD DS SW WZ ZC CD DL LQ QB BX XJ JD DY YD

Fig. 4. Salt marsh loss caused by reclamation in different locations along the coast of Jiangsu Province between 1987 and 2013 (Left Y axis: areas of salt marsh loss caused by reclamation; right Y axis: percent of total salt marsh loss caused by reclamation. SX, Sheyang-Xinyang; XD, Xinyang-Doulong; DS, Doulong-Simaoyou; SW, Simaoyou-Wanggang; WZ, Wanggang-Zhugang; ZC, Zhugang-Chuandong; CD, Chuandong-Dongtai; DL, Dongtai-Liangduo; LQ, Liangduo-Qionggang; QB, Qionggang-Beiling; BX, Beiling-Xiaoyang; XJ, Xiaoyang-Jueju; JD, Jueju-Dongling; DY, Dongling-Yaowang; YD, Yaowang-Dongzao. The distribution of each location is in Fig.1. Data from Sun et al., 2015a).

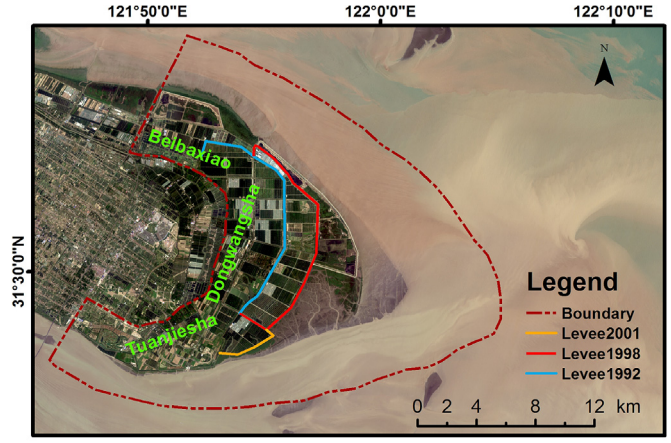

Fig. 5. Satellite image showing the area reclaimed through three reclamation levees built consecutively in Chongming Dongtan, Shanghai City (Landsat 8 Operational Land Imager data from the United States Geological Survey, acquired on 08/03/2015; Chongming Dongtan includes three parts: Tuanjiesha (south), Dongwangsha (east), and Beibaxiao (north)).

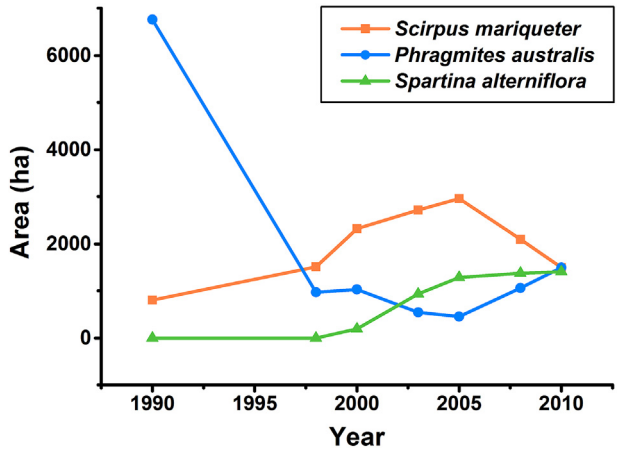

Fig. 6. Salt marsh dynamics in Chongming Dongtan, Shanghai City (Data from Huang, 2009).

3.2. Climate change and coastal erosion

Since 1960, the climate in China has become generally warmer, with an average increase of annual temperature of 1.2 °C, and become dryer, especially in the northeastern China where salt marshes are mostly concentrated (Piao et al., 2010). Climate change, combined with excessive freshwater withdrawals by a growing population and agricultural and industrial demands (Zhao et al., 2009), contributed to more evaporation and reduced fresh water supply to coastal wetlands. As a result, soil salinity increased in coastal regions. Salinity not only controls the osmotic pressure experienced by salt marsh plants, but also affects seed germination (Noe and Zedler, 2000), photosynthesis and growth of salt marsh plants (Rozema and Diggelen, 1991). Though vegetation in salt marsh is salt-tolerant, each species has a tolerance limit to salinity. For example, *Suaeda heteroptera* Kitag, creating the famous red beach in the Liao River Delta, suffered a loss due to increasing salinity (Wang and Li, 2006). In addition, the construction of a barrage in the Liao River estuary reduced the fresh water supply to coastal salt marshes, further accelerating the increase in salinity (Song and Lv, 2009) and loss of salt marshes. While increased salinity can lead to loss of salt marshes, it can also result in changes in salt marsh vegetation through two separate mechanisms. One is change in salt marsh vegetation to more salt-tolerant species. The other is transition from unvegetated high intertidal flats to salt marshes similar to "landward migration" due to sea level rise. The rate of salt marsh change accelerates with increasing salinity, depended on other modulating factors such as tide height and frequency, evaporation, precipitation and other climatic factors (Wang et al., 2012).

Coastal erosion is also an important driver of salt marsh loss. Land subsidence decreases the elevation of salt marsh area. Combined with sea level rise and fixed landward boundaries like seawalls, land subsidence accelerates the inundation of salt marshes, causing salt marsh damage. Consequently, previous salt marsh areas will become unvegetated flats or will be inundated by seawater. This leads to the area being subjected to coastal erosion. Indeed, coastal areas across China have experienced significant land subsidence due to human activities, especially in major river-delta regions (i.e., the Yangtze River Delta, the Yellow River Delta, and the Pearl River Delta, Yin et al., 2012). Subsidence rates at aquaculture facilities had been estimated as high as 250 mm $year^{-1}$ between 2007 and 2011 in the Yellow River coastal area, probably due to groundwater pumping (Higgins et al., 2013). These rates exceed local and global average sea level rise by nearly 2 orders of magnitude and aggravate coastal erosion. Coastal erosion was extremely severe in the Yellow River Delta (Zhang and Li, 2008), where a total of 22,270 ha salt marshes and a total of 34,350 ha tidal flats were lost when flooded by seawater due to erosion and apparent sea level rise (Liu, 2013).

3.3. Invasive plant species

Since it was first introduced in China in 1979, *Spartina alterniflora* has expanded to occupy extensive areas due to its broad ecological niche and lack of natural controlling agents (e.g. herbivores and pathogens). As a result, the area of *Spartina alterniflora* is now larger than the area of *Spartina anglica* introduced in 1963. *Spartina alterniflora* has been increasing since it was first observed in 1982 in the middle coast of Jiangsu Province (Xu and Zhuo, 1985), expanding at rate of about 9.6 $km^2\ yr^{-1}$ (Liu et al., 2004). However, the pace of coastal reclamation was faster than the growth of *Spartina alterniflora*, leading to a decline in the area occupied by *Spartina alterniflora* in some regions

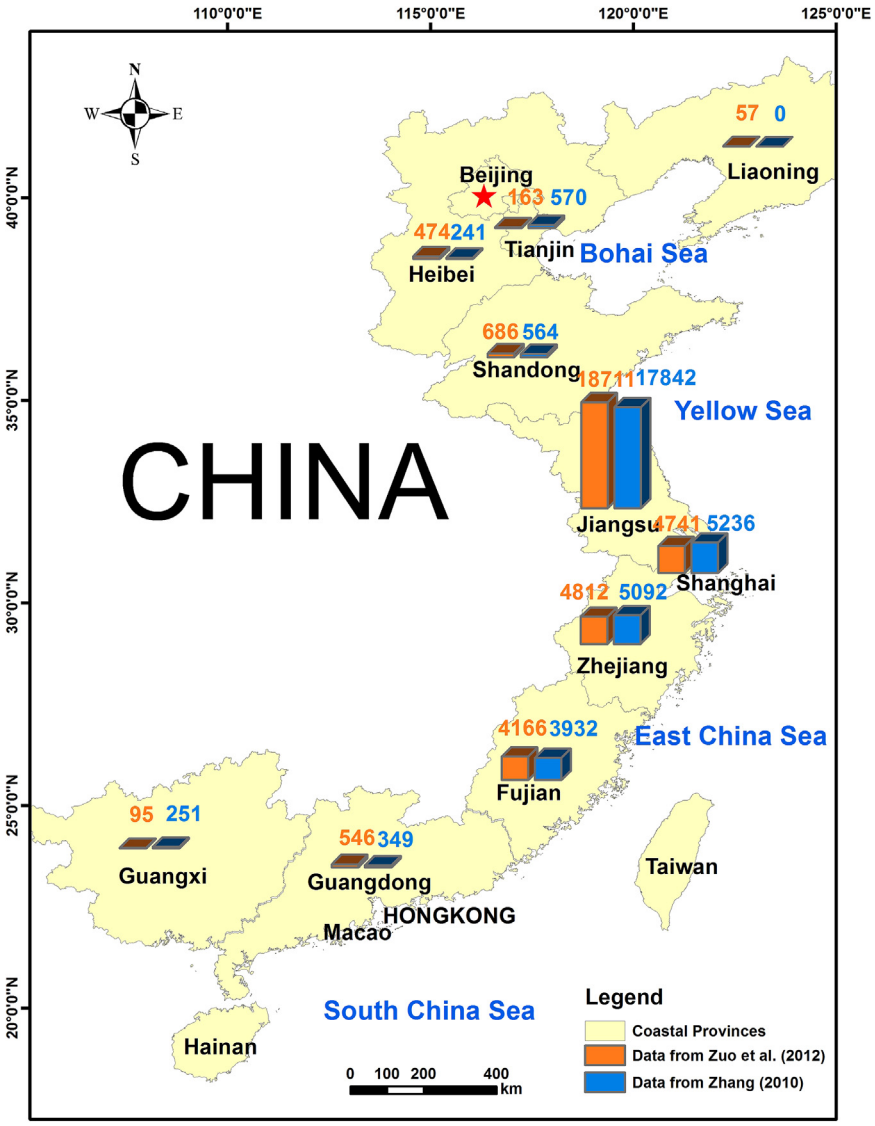

Fig. 7. Spatial distribution of Spartina alterniflora along coastal China in 2007 (Data from Zhang, 2010 and Zuo et al., 2012).

(Zhu and Gao, 2014). *Spatina alterniflora* was introduced into Chongming Dongtan in 1995 (Wang, 2011). After 2005 *Spartina alterniflora* adapted to the local environment and expanded to the middle and lower intertidal flats, leading to a sharp competitive exclusion of *Scirpus mariqueter* and a change of composition of different salt marsh species (Fig. 6). In Chongming Dongtan, *Spartina alterniflora* invasion indeed led to severe loss of native salt marsh *Scirpus mariqueter*, which requires control of *Spartina alterniflora* to conserve native salt marshes for providing birds with more habitats (Tang, 2016). Though *Spartina alterniflora* led to native salt marshes decrease, total salt marsh extent might increase due to its fast expansion. Thus, the distribution and extent of invasive species are of great importance to understand its effects on local ecosystems. Use of Landsat TM and CBERS-1 satellite images to assess the distribution of *Spartina alterniflora* yielded a total area of about 34,300 ha in 2007 (Zhang, 2010; Zuo et al., 2012), mostly concentrated in the central coast of China, with much smaller extents in northern and southern coastal provinces. Specifically, half of the total area of *Spartina alterniflora* was found in the Jiangsu Province, with other three city/provinces Shanghai, Zhejiang and Fujian also having significant distributions (Fig. 7). The areas of *Spartina alterniflora* showing in Fig. 7 for each province differed somewhat between the two studies, which may be attributable to the different remote sensing data and classification methods used in these studies (Zhang, 2010; Zuo et al., 2012).

3.4. Natural competition and succession of salt marsh vegetation

Scirpus mariqueter, growing in the lower intertidal flats, is not only the pioneer plant in Chongming Dongtan salt marsh area but also is an endemic plant to China, which provides habitat and food for migratory birds in the Yangtze River estuary (Wang, 2007). Deposition of sediments from the Yangtze River resulted in the expansion of Chongming Dongtan, with an increased elevation of low intertidal flats allowing colonization by *Scirpus mariqueter*, which increased until 2005 (Fig. 6). However, due to reduced sediments inputs from the Yangtze River (Liu et al., 2010) and invasion by *Spartina alterniflora*, *Scirpus mariqueter* was decreasing after 2005.

Jiuduansha, in the southeast of Chongming Dongtan (Fig. 1d), is continuously growing in elevation and extent since the mid-1950s (Yang et al., 2006), due to increased sediments delivery, resulting in its transition from aquatic ecosystem to terrestrial ecosystem. The pioneer salt marsh plant *Scirpus mariqueter* colonized the low unvegetated intertidal flats in the late 1980s and *Phragmites australis* appeared in the 1990s as elevation increased (Tang and Lu, 2003). This succession sequence led to the formation of a salt marsh ecosystem in Jiuduansha. *Spartina alterniflora* was introduced in the late 1990s to compensate the

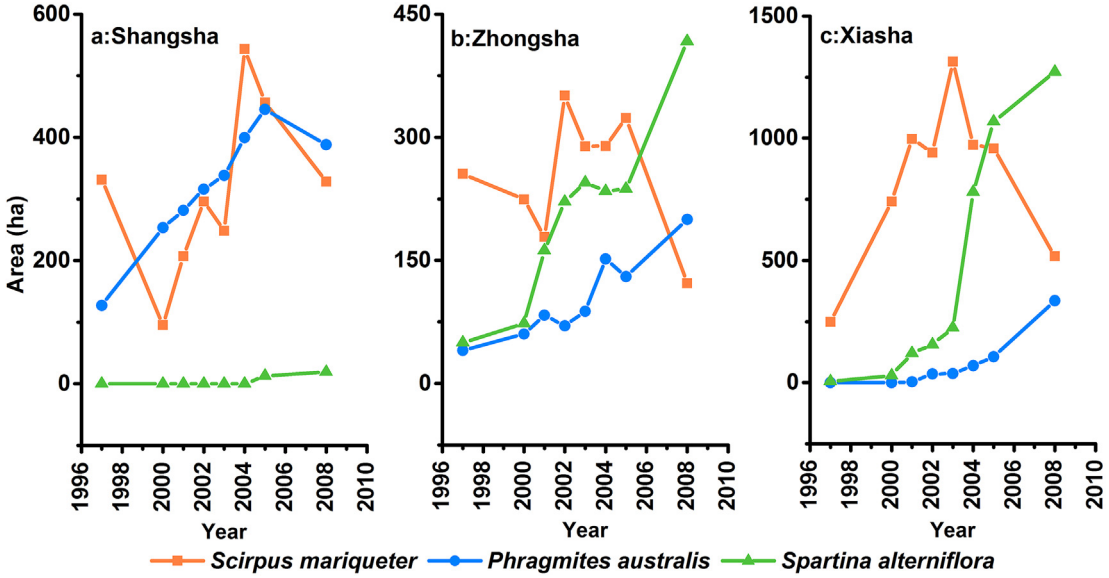

Fig. 8. Salt marsh dynamics in Jiuduansha, Shanghai City (a, Shangsha; b, Zhongsha and c, Xiasha. Data from Huang, 2009).

loss of tidal flats reclaimed to build the Pudong International Airport in Shanghai. This exotic salt marsh plant accelerated the sediment deposition, leading to rapid growths in extent and elevation in Jiuduansha, therefore, resulting in expansion of salt marshes and the vegetation succession in Jiuduansha (Tang and Lu, 2003).

The three dominant salt marsh plant species mentioned above experienced different dynamics across Jiuduansha (Fig. 8a–c, Huang, 2009). Jiuduansha is composed of three parts: Shangsha (the upper intertidal), Zhongsha (the middle intertidal) and Xiasha (the lower intertidal) regions. By visually comparing the plots of Fig. 8a–c, it is observed that the area dominated by the native plants *Scirpus mariqueter* presented some considerable fluctuations, whereas the areas dominated by the other two plants (*Spartina alterniflora* and *Phragmites australis*) exhibited upward trends. Specifically, in Shangsha (Fig. 8a), the native plants *Scirpus mariqueter* and *Phragmites australis* dominated the salt marshes between 1997 and 2008. Due to the Yangtze Estuary Deepwater Channel finished in 1999, sediment deposition accelerated in Shangsha, leading to a rapid growth in elevation and extent. Thus, *Scirpus mariqueter* and *Phragmites australis* totally increased. The exotic plant *Spartina alterniflora* was first planted in this region in 2004 and was adapting to the local conditions with slow expansion. *Scirpus mariqueter* was the dominant plant in 1997 in Zhongsha due to low mean elevation (Fig. 8b). Total 40 ha *Phragmites australis* and 50 ha *Spartina alterniflora* were planted in 1997, only forming small local patches. While mean elevation and extent in Zhongsha increased due to sediment accumulation, the three salt marsh species increased at different rates. *Spartina alterniflora* competitively excluded *Scirpus mariqueter* in lower elevation areas and competed with *Phragmites australis* in higher elevation areas. Thus, in 2008 *Spartina alterniflora* became the dominant plant, *Phragmites australis* follwed, and then *Scirpus mariqueter*. In Xiasha (Fig. 8c), only *Scirpus mariqueter* was present in 1997. It experienced a fast expansion by colonizing new unvegetated intertidal flats until 2003. Only 5 ha *Spartina alterniflora* was planted in 1997. Due to low mean elevation and strong wind and wave action, *Spartina alterniflora* was increasing slowly as small patches during the previous years after plantation. After 2003, *Spartina alterniflora* formed a separate dominant community with rapid expansion, which was caused by two reasons. On the one hand, *Spartina alterniflora* expanded into higher elevation areas with increased marsh elevation, while on the other, *Spartina alterniflora* invaded into new *Scirpus mariqueter* marshes that were colonized on unvegetated intertidal flats due to increased sediment deposition. The area covered by *Phragmites australis* was increasing, but still presented in small area by 2008 due to its overall low mean elevation.

Although the constructions of more than 50,000 dams in the recent 50 years in its tributaries together with reduced precipitation and soil conservation programs have led to a decline in sediment supply from the Yangtze River (Yang et al., 2015), both the total area and the mean elevation of Jiuduansha salt marsh have increased (Chen et al., 2011). In addition, the salt marsh vegetation is gradually becoming more diverse. While Chongming Island is the first generation and Changxing Island and Hengsha Island are the second generation, Jiuduansha is the third-generation alluvial sandbank in the Yangtze River estuary (Yang et al., 2006). Local people call Jiuduansha as the third generation of Chongming Island, referring to its rapid elevation acceleration and fast vegetation ecosystem evolution. Accordingly, in the absence of anticipated disturbances, the total area of Jiuduansha salt marsh is expected to continue to increase.

3.5. Summary of salt marsh dynamics

Coastal reclamation emerges in our review as the dominant cause for salt marsh loss across China, with four main reclamation periods since the 1950s (China Water Conservancy Association Mud Flat Development Committee, 2000; Sun et al., 2010). In the 1950s, a campaign to increase salt production led to reclamation of salt marshes to convert them into salt pans. From the 1960s–1970s, conversion to agricultural land to meet the growing demand for food from a growing population was the main driver of salt marsh reclamation. In the 1980s and early 1990s, conversion to aquaculture ponds became the dominant driver of salt marsh reclamation, especially in the north of China. Since the late 1990s, urban, industrial and port expansion have been the dominant driver of salt marsh reclamation. The timing of these four drivers of reclamation differed somewhat across provinces and multiple drivers often operated concurrently in some areas. Reclamation projects reached a maximum in 2009, when 17,888 ha were reclaimed across China. In addition to coastal reclamation, coastal erosion leads to further loss of salt marshes. Climate change has either negative or positive effects on salt marsh extent depending on specific local conditions. Invasive species and natural competition and vegetation succession mostly influence the composition of different salt marsh plants, and have only resulted in increased salt marsh extent, because of the difference in the capacities of the different species as engineering species able to accrete sediments, in some areas like Jiuduansha.

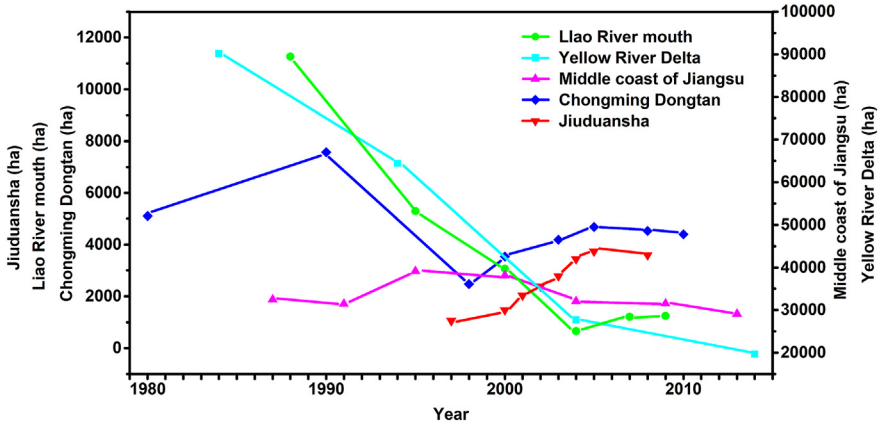

Fig. 9. Salt marsh trends in dominant regions along coastal China (Data from Shen et al., 2006; Huang, 2009; Jia et al., 2015; Liu et al., 2015; Sun et al., 2015a).

As presented above, the variations of salt marshes from the 1980s–2010s in these five coastal regions differed widely (Fig. 9). Salt marshes suffered great losses, with a net areal reduction of 89.0% and 78.2% in the Liao River Delta and the Yellow River Delta, respectively. In the middle coast of Jiangsu and Chongming Dongtan, large areas of salt marshes were reclaimed. Meanwhile, sedimentations have led to the formation of tidal flat and then colonized by new salt marshes. As a result, a modest decrease occurred, with a net areal reduction of 10.3% and 13.7% in the middle coast of Jiangsu and Chongming Dongtan, respectively. The total salt marsh area in Jiuduansha wetland has increased from 1059 ha in 1997 to 3601 ha in 2008 due to continues sedimentation. Overall, in these five coastal regions studied from the 1980s–2010s, the salt marsh areas decreased from 140,065 ha to 58,046 ha, a net reduction of 59%, though the rate of salt marsh loss slowed down after the year 2000.

Salt marshes in other nations have also suffered great losses. The average salt marsh loss in New England has been report to be 37% over the last 200 years (Bromberg and Bertness, 2005). Salt marshes in Jamaica Bay decreased by an average 38% since 1974, with smaller areas losing up to 78% of salt marshes (Hartig et al., 2002). In Chesapeake Bay, salt marshes suffered 50% of loss over the past 200 years, but the loss rate slowed down after protective legislation enacted in the 1970s (Dahl, 1990). Drivers of salt marsh loss vary across the world, but reclamation for agricultural, industrial and urban development and sea level rise with climate change are the common drivers of salt marsh loss all over the world (Hartig et al., 2002; Wilson et al., 2007; Gedan et al., 2009; Kennish, 2013). Yet for a specific region the main driver of salt marsh loss may differ, such as insufficient sediment supply in Galveston Bay, Texas (Ravens et al., 2009). Invasive species also resulted in salt marsh changes in other areas. In north America *Phragmites australis* invaded into native salt marshes, leading to great changes in salt marsh ecosystems, such as Chesapeake Bay (Rice and Rooth, 2000).

4. Factors affecting future trends in salt marsh extent

4.1. Sea level rise

Sea level rise is a major threat to coastal salt marsh across China (Ma et al., 2014). The salt marshes in the old Yellow River Delta are vulnerable to sea level rise. The middle coast of Jiangsu is expected to face a high sea level rise vulnerability due to severe subsidence and low elevation along with a large tidal range (Yin et al., 2012). In the Yangtze River Delta, without consideration of sedimentation, Tian et al. (2010) predicted that 7% and 14% of salt marshes in Chongming Dongtan could be lost as a result of a 0.88 m of sea level rise by 2050 and by 2100. Under the present sea level rise of 0.3 m (low level) by 2100, Ge et al. (2016) predicted that salt marshes in Chongming Dongtan and Jiuduansha would increase by 6% in the short term by 2025, 4% in the medium term by 2050, and 3% in the long term by 2100, respectively. However, under the IPCC RCP (Representative Concentration Pathway) 8.5 max scenario of 0.98 m (high level) by 2100, salt marshes in both Chongming Dongtan and Jiuduansha would suffer a risk of decrease by 2%, 5% and 9% in the short, medium, and long term, respectively (Ge et al., 2016). These predictions consider both the sedimentary and salt marsh processes, which provide more reliable estimates than those only based on sea level rise, since salt marshes can adjust their accretion, within some limits, to sea level rise (Kirwan et al., 2016). In addition, salt marshes can migrate, where no barriers are present, upslope with increasing sea level rise, substituting terrestrial vegetation. They can also extend seaward where high rates of mudflat accretion exceed the rate of submergence (Ge et al., 2016). Management actions to accelerate mudflat expansion and the removal of seawalls, currently extending over 60% of the shoreline of China (Ma et al., 2014) and preventing landward migration of salt marshes, could help salt marsh adapt to sea level rise.

4.2. Reclamation pressure

Coastal reclamation has led to great loss of salt marshes in the past and it will continue to affect future salt marsh extent. Further reclamation is expected to include more than 578,000 ha reclamation areas scheduled by 2020, under planned marine zoning in coastal provinces approved by the State Council in 2001, which means 28,900 ha of coastal habitats would be reclaimed per year until 2020. However, in recent years, under the control of governments and administration sectors, reclamation projects have decreased evidently. In 2016, China approved Shoreline Protection and Utilization Management Practices, Reclamation Management and Control Approaches, and Marine Inspector Program to improve integrated ocean management, which set stricter regulations on reclamation. For example, the State Oceanic Administration in July 2017 ordered the suspension of man-made island reclamation occupying nearly 90 ha coastal area for Sanya new airport in Hainan Province, due to its substandard environmental assessment report. The State Council (2018) issued a public announcement about strict control of reclamation to enhance coastal wetlands protection. The announcement totally banned new reclamation projects except national strategic projects, commanded proper treatment with old reclamation projects, and highly emphasized on ecological restoration, helping to relieve reclamation pressure and effects on coastal wetlands in the future.

4.3. Environmental pollution

Freshwater inputs often carry nutrients, chemicals and heavy metals to salt marshes, while groundwater often acts as a more important source of nitrogen to salt marsh than freshwater runoff (Valiela and

Table 2
Nature reserves of coastal wetlands in main salt marsh areas.

Name	Year	Main protection targets	Salt marsh
Yancheng Wetland	1983	*Grus japonensis*; mudflat wetland ecosystem	PA、SS、SA
The Liao River Mouth	1985	Rare waterfowls; coastal wetland ecosystem	PA、SS
The Yellow River Delta	1990	Estuarine wetland ecosystem; rare birds	PA、TC、SS、SA
Chongming Dongtan	1998	Migratory birds; wetland ecosystem	PA、SM、SA
Jiuduansha Wetland	2000	Estuarine sandbar; birds	PA、SM、SA

(PA: *Phragmites australis*; SS: *Suaeda salsa*; SM: *Scirpus mariqueter*; SA: *Spartina alternilfora*; TC: *Tamarix chinensix*).

Teal, 1979). Salt marshes have a high capacity to remove nutrients, such as nitrogen and phosphorus, derived from house-hold, industrial and agricultural waste and sewage. However, if nutrient concentrations exceed the maximum levels tolerated by salt marshes, then it will be a threat to salt marsh ecosystems, as demonstrated by the experimental fragment distribution of salt marshes derived from long-term nutrient enrichment (Deegan et al., 2012). In addition, eutrophication leads to increased algal mats proliferation in salt marshes, reducing its resilience to sea level rise (Wasson et al., 2017). As severe eutrophication is widespread in Chinese coastal waters (Jiang et al., 2018), nutrient inputs in excess to the thresholds tolerated by salt marshes represent a threat to their conservation.

5. Salt marsh management and recommendations

5.1. Existing management practices

China has developed a system of protected areas to conserve coastal wetlands. Most salt marsh areas are now included in nature reserves (Table 2), where management plans are in place to protect and restore salt marsh wetlands. In the Yellow River Delta nature reserve, strict land use control is applied to protect wetlands, distributed in 8 different planning zones. Ecological conservation zones, occupying 37.5% of total reserve area, are aimed to protect natural wetland ecosystems such as *Phragmites australis* and important habitats for birds. Wetland restoration zones, covering 13.4% of total area, are designed to restore degraded wetlands. Apart from areas that have undergone wetlands restoration, areas with enough freshwater inputs are also included in restoration zones, as elevated salinity is often a cause of salt marsh degradation in the Yellow River Delta (Sun et al., 2011). In Chongming Dongtan, where *Spartina alterniflora* encroaches into *Scirpus mariqueter* and *Phragmites australis* vegetation, efforts to control the rapid expansion of *Spartina alternilfora* are in place to protect habitats for migratory birds (Tang, 2016). Ecotourism also provides a sustainable way to use and manage wetlands, as it provides the public with the opportunity to appreciate the ecological values of coastal wetlands, thereby increasing their awareness and support to management efforts. For instance, public appreciation of the values of red beach *Suaeda heteroptera* Kitag salt marsh, which represents a major tourism attraction in the Liao River Delta, catalyzed efforts to recover this salt marsh ecosystem. Salt marsh wetlands are key habitats for many birds, thereby providing sites for bird watching, a main activity in nature reserves in the Yellow River Delta, Yancheng and Chongming Dongtan (Table 2), also helps gain support from the public for salt marsh management.

The State Forestry Administration defined, in 2013, a “redline” for wetland protection establishing a minimum of 54,170,000 ha, including 5,800,000 ha coastal wetlands in China by 2020 (State Forestry Administration, 2013). In 2012 the State Oceanic Administration commanded the provinces around the Bohai Sea to establish a marine ecological redline (State Oceanic Administration, 2012) and encouraged all coastal provinces to define marine ecological redlines and specify the measures to meet them under the direction of Technical Guidelines for National Marine Ecological Red Line Demarcation in 2016 (State Oceanic Administration, 2016). The State Oceanic Administration published an Implementation Plan of Marine Ecological Civilization Construction (2015–2020), providing a time line and a road map guiding oceanic sectors to promote marine ecological civilization construction (State Oceanic Administration, 2015). The plan includes 10 items within the period of 2016–2020, that is, planning guidance and constraint, total amount control and redline control, marine resource allocation and management, marine environment monitoring and pollution prevention, marine ecological protection and restoration, maritime law enforcement, governance performance and responsibility assessment, marine scientific innovation capability improvement, marine personnel training, and public education and participation. A number of national projects aiming at restoring marine ecosystems –such as Blue Bay, Silver Beach, Mangrove in South and *Tamarix* in North, and Ecological Island– are included in the Implementation Plan (2015–2020).

5.2. Issues and recommendations

The threat reclamation poses to salt marsh conservation is partially associated with the lack of national legislation to prevent wetland loss, leaving reclamation to be managed by regulations and policies, which is lack of the enforcement power of laws in China. Indeed, the Land Administration Law of China regards coastal wetlands as unused land. Hence, local governments in coastal regions prefer to reclaim coastal wetlands as the fastest, cheapest and most effective way to increase land for agriculture and development uses under current laws and policies. It is estimated that the cost of converting coastal wetland to cultivated land is 400–667 RMB (Chinese Yuan) per ha, compared to 2000 RMB per ha of cultivated land and 33,333–666,667 RMB per ha of land for construction (Lei et al., 2017), depending on the specific construction land use. Thus, reclamation of coastal wetlands provides an attractive source of revenue to local governments. Whereas existing regulations on the Protection of Wetland Management can be regarded as the foundation of national legislation, only the establishment of legal basis for wetland protection would confine reclamation activities and help conserve salt marshes in China.

In addition to a national law protecting wetlands, effective measures must be taken to guarantee “zero” loss of coastal wetlands, if the 5,800,000 ha of coastal wetland included within the 54,170,000 ha of the Ecological Redline agreed by the State Council is to be met. A national strategy to provide land for various uses is also required to avoid reclamation pressures on salt marshes, including ecological compensation policies and costs to those delivering pollution to salt marshes. Moreover, assessing wetland management practices as a part of the government's performance assessment can encourage local governments to protect and restore coastal wetlands as much as possible.

Beyond enforcement, improving public awareness and community participation in wetland protection may further help avoid salt marsh losses (Polajnar, 2008). The experience of other nations has proven that participatory mechanisms (involving local communities and residents) can be highly effective in enhancing salt marsh conservation. Encouraging local communities to participate in wetland management in China can help educate the public on the importance of coastal wetland and gain support for salt marsh protection. The State Council has realized the importance of public participation and indicated it in the new announcement (State Council, 2018).

6. Conclusions

Conservation objectives in China require accurate figures on salt marsh extent to trace possible improvement or degradation. Historical extent and loss of salt marsh can identify feasible locations for

restoration and set proper targets for amounts in the restoration projects (Van Dyke and Wasson, 2005). Hence, an accurate description of salt marsh extent is important to the rigorous assessment of the progress made by conservation and environmental agencies towards meeting biodiversity objectives. Additionally, regular monitoring and assessment of salt marshes can further help understand, protect and restore salt marshes, which has been carried out in the Wadden Sea area (Esselink et al., 2017). The findings of this work allow a comprehensive evaluation of salt marsh dynamics in China, thus providing the foundation for an improved appraisal of losses and gains at a nationwide level. In particular, it was found that the trends of coastal salt marshes in China differed greatly among the five reported regions, with an overall loss of 59% salt marsh area from the 1980s–2010s. The rate of salt marsh loss slowed down after the year 2000. The loss of salt marsh is the result of natural and anthropogenic activities. Coastal reclamation and infrastructural construction accounted for most of the degradation and loss of salt marshes. Coastal erosion further accelerated salt marsh loss. The invasive species *Spartina alterniflora*, to some extent, increased salt marsh area but it made local salt marsh plants decrease. Climate change, natural competition and succession of salt marsh vegetation resulted in differential influences on salt marshes. Salt marshes in China still face various threats, including sea level rise, reclamation pressure and environmental pollution. Although China has implemented several measures to protect and restore salt marshes, such as setting up protected areas, drawing marine ecological redlines and enacting strict reclamation regulations, the actual outcomes are less satisfactory than expected. The establishment of legal basis for wetland protection, more effective salt marsh management and enforcement measures, and enhancement of public awareness and community participation are recommended to further help conserve salt marshes. Currently, China is vigorously developing a blue carbon program of mitigation and adaptation for climate change, which supports the need for salt marsh conservation.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecss.2018.09.015.

Declaration of interest

None.

Contributors

Conceived and designed the analysis: J.W., C.D., J.G.; searched and collected data and materials: J.G., M.L. X.Z.; all authors analyzed, interpreted the data and results, and edited the manuscript; wrote the manuscript: J.G, C.D., J.W.

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